# Influence of land use on fine sediment in salmonid spawning gravels within the Russian River Basin, California

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# Influence of land use on fine sediment in salmonid spawning gravels within the Russian River Basin, California

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**Abstract:** Relationships between land use or land cover and embeddedness, a measure of fine sediment in spawning gravels, were examined at multiple scales across 54 streams in the Russian River Basin, California. The results suggest that coarse-scale measures of watershed land use can explain a large proportion of the variability in embeddedness and that the explanatory power of this relationship increases with watershed size. Agricultural and urban land uses and road density were positively associated with embeddedness, while the opposite was true for forest cover. The ability of land use and land cover to predict embeddedness varied among five zones of influence, with the greatest explanatory power occurring at the entire-watershed scale. Land use within a more restricted riparian corridor generally did not relate to embeddedness, suggesting that reach-scale riparian protection or restoration will have little influence on levels of fine sediment. The explanatory power of these models was greater when conducted among a subset of the largest watersheds (maximum $r^2$ = 0.73) than among the smallest watersheds (maximum $r^2$ = 0.46).

**Résumé :** Nous avons examiné à plusieurs échelles les relations entre l'utilisation des terres et la couverture végétale, d'une part, et le colmatage du substrat, une mesure des sédiments fins dans les graviers de reproduction, d'autre part, dans 54 cours d'eau du bassin versant de la rivière Russian, Californie. Nos résultats indiquent que des évaluations de l'utilisation des terres dans le bassin versant à une échelle grossière peuvent expliquer une proportion importante de la variabilité du colmatage et que le pouvoir explicatif de cette relation augmente en fonction de la taille du bassin versant. Il y a une association positive entre les utilisations urbaine et agricole des terres et la densité des routes, d'une part, et le colmatage, d'autre part, alors que la relation est inverse dans le cas de la couverture forestière. Le potentiel de l'utilisation des terres et de la couverture végétale pour prédire le colmatage varie en fonction des cinq zones d'influence et le potentiel maximal se manifeste à l'échelle du bassin versant entier. Il n'y a pas généralement de corrélation entre l'utilisation des terres sur un corridor plus étroit le long des rives et le colmatage, ce qui laisse croire que la restauration ou la protection des rives au niveau de la section du cours d'eau aura peu d'influence sur les quantités de sédiments fins. Le pouvoir explicatif de ces modèles est plus grand lorsqu'il s'applique à un sous-ensemble des bassins versants les plus grands ($r^2$ maximal = 0,73) plutôt qu'aux plus petits bassins versants ($r^2$ maximal = 0,46).

[Traduit par la Rédaction]

## Introduction

In California, coastal watersheds once supported prodigious runs of six species of anadromous salmonids: coho (*Oncorhynchus kisutch*), Chinook (*Oncorhynchus tshawytscha*), pink (*Oncorhynchus gorbuscha*), and chum salmon (*Oncorhynchus keta*), and steelhead (*Oncorhynchus mykiss*) and sea-run coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) (Moyle 2002). At present, many of these runs have been extirpated from coastal drainages (e.g., pink and chum salmon from the Russian River Basin), while those that remain are generally listed as threatened or endangered under the Federal Endangered Species Act (Mills et al. 1997; Busby et al. 2000; Weitkamp et al. 2000).

Because degradation of freshwater habitat is one of the key factors leading to the decline of anadromous fish along the Pacific coast of the United States (Nehlsen et al. 1991; National Research Council 1996; Nehlsen 1997), resources and attention devoted to stream restoration have greatly increased, and millions of dollars are currently being spent to restore fish habitat (Roper et al. 1997; Roni et al. 2002). In particular, sedimentation has been identified as one possible

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agent degrading freshwater ecosystems and limiting the persistence and recovery of salmonid populations. High levels of fine sediment (<2 mm diameter) in spawning gravels are correlated with low survival of salmonid eggs and alevins (Everest et al. 1987; Reiser and White 1988; Kondolf 2000). Further, high levels of fine sediment can simplify bed features, reduce cover and populations of macroinvertebrates, and fill pools (Henley et al. 2000; McIntosh et al. 2000; Suttle et al. 2004).

Land-use activities that alter or replace native vegetation are considered key drivers leading to increased sediment production and delivery to streams (Waters 1995; Pimentel and Kounang 1998). Numerous studies have found that land-use activities such as agriculture, grazing, roads, and urban development can lead to elevated sediment production, both directly (e.g., rill and gully erosion) and indirectly (reduced infiltration leading to higher peak flows and channel degradation) (Swanson and Dyrness 1975; Montgomery 1994; Pizzuto et al. 2000). While these studies identify the mechanisms by which land-use activities produce sediment, methods to predict characteristics of in-stream sediment (e.g., magnitude of fluxes, grain size) based on patterns of watershed land use remain limited in scope, scale, and explanatory power (Nilsson et al. 2003). Given these limitations, it has been suggested that a promising alternative is to build empirical relationships between land use and observed sediment fluxes or concentrations (Nilsson et al. 2003).

An important issue that has emerged with empirical modeling is the spatial scaling of variables (Allan et al. 1997; Allan and Johnson 1997; Strayer et al. 2003). Recent research has explored linkages between land use and aquatic habitat at various scales (Hunsaker and Levine 1995; Lammert and Allan 1999; Strayer et al. 2003), including levels of fine sediment in streams (Richards et al. 1996; Wohl and Carline 1996; Snyder et al. 2003) and indices of salmonid abundance (Paulsen and Fisher 2001; Pess et al. 2002; Regetz 2003). These studies have reached conflicting conclusions regarding the spatial scale (or zone of influence) at which land use can predict effectively the ecological response of the stream reach. For example, several studies have concluded that land use within the local area (e.g., the riparian corridor surrounding or immediately upstream of a site) has a greater influence on the freshwater ecosystem than land use within the entire watershed (Jones et al. 1999; Lammert and Allan 1999; Sponseller et al. 2001), while other studies have come to the opposite conclusion (Omernik et al. 1981; Roth et al. 1996; Wang et al. 1997). Differences in the size of watersheds examined by these studies may have contributed to their different conclusions on the effects of land use on aquatic ecosystems. By elucidating the scales at which land use influences habitat, research on land use across scales can help managers target their restoration and management efforts to the appropriate scale.

Nearly all research on the influence of land use across scales has been concentrated in the eastern and midwestern United States (US). Further, in the western US, the research available to inform restoration programs for anadromous fish has been largely conducted in the conifer-dominated watersheds of the Pacific Northwest. Conversely, very little research on salmonid habitat has occurred within the hardwood-dominated, Mediterranean-climate watersheds of northern and central California. These watersheds have much more rugged topography than those studied in the eastern and midwestern US and different vegetation and hydrologic regimes and more varied land uses than those found in the Pacific Northwest. Therefore, generalizations about scales of influence and the role of land use developed in other regions may not be able to be directly transferred to Mediterranean-climate watersheds. An understanding of the basic processes that shape habitat, and the scales at which they operate, can help the adaptation of restoration strategies to this region.

This paper examines the empirical relationship between land use and land cover (LULC) and the level of fine sediment in spawning gravels across 54 streams in a Mediterranean-climate basin in northern California. Specifically, we ask the following questions. (*i*) Can readily available data on land use help explain the patterns of fine sediment found in streams across this basin? (*ii*) Within what zone of influence, ranging from the local riparian corridor to the entire watershed, does land use have the most explanatory power for predicting levels of fine sediment and what accounts for these differences in explanatory power? (*iii*) What is the effect of watershed size on the predictive power of these empirical relationships? This last question is motivated by the work of Strayer et al. (2003), who concluded that the ability of LULC patterns to explain in-stream variables became weaker in smaller watersheds (e.g., <1000 ha). Thus we examine the influence of spatial scale in two different ways: within watersheds (i.e., zone of influence) and across watersheds (e.g., large watersheds vs. small watersheds).

## Materials and methods

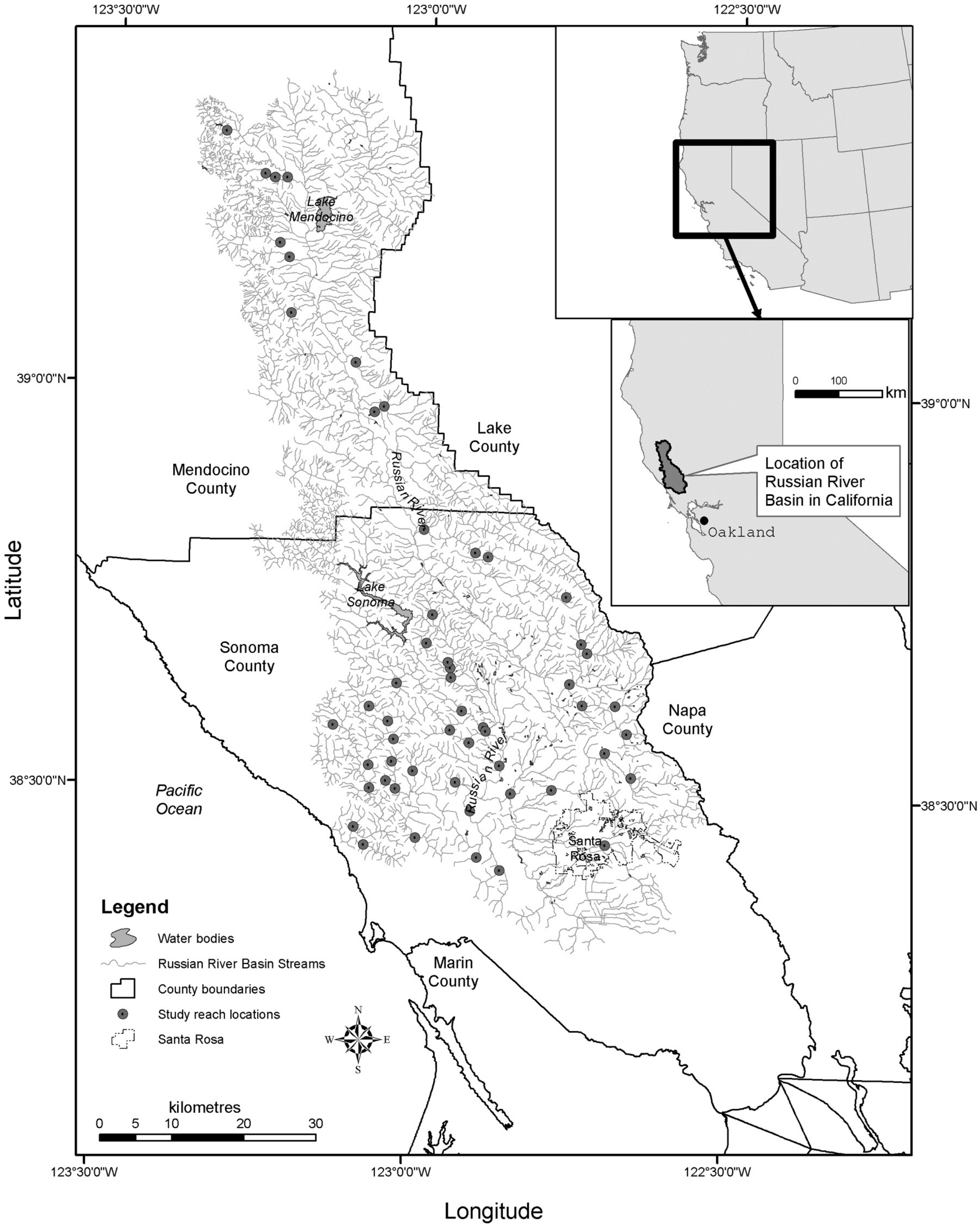
### Study region

The Russian River Basin is located in northwestern California within Sonoma and Mendocino counties (Fig. 1). The basin is currently listed as impaired by sediment under Section 303(*d*) of the Federal Clean Water Act. The basin also provides habitat for several species of anadromous fish, including Chinook and coho salmon and steelhead trout. All three runs are currently listed under the Federal Endangered Species Act as being threatened or endangered.

The mainstem Russian River flows for approximately 175 km within a 3850-km$^2$ basin that is underlain primarily by the Jurassic–Cretaceous age Franciscan Formation. Elevations range from sea level to 1325 m. The basin has a Mediterranean climate with cool, wet winters and hot, dry summers (Gasith and Resh 1999); the mean annual rainfall ranges from 69 cm to as high as 216 cm across the subbasins, with the majority of precipitation between December and March and little or no rain between May and October.

The majority of the basin is dominated by hardwood forests, oak savannas, chaparral, and grasslands. Conifer-dominated forests occur near the coast and intermittently throughout the basin on north-facing slopes. Land use is varied and includes vineyards, orchards, and other agriculture, sheep and cattle grazing, timber harvest, and urban, suburban, and low-density rural residential housing. Vineyards, urban areas, and suburban developments dominate the valley floors. Currently, there are high rates of land-use change on the hillslopes with conversion from natural vegetation to vineyard and low-density residential development (Meren-

**Fig. 1.** Location of 54 stream reaches in the Russian River Basin (California, USA) surveyed for levels of fine sediment in spawning gravels.




lender et al. 1998; Heaton and Merenlender 2000; Merenlender 2000).

In-stream habitat data were collected by the California Department of Fish and Game (CDFG) in the Russian River Basin between 1997 and 2000 (Fig. 1). Field crews recorded the concentration or level of fine sediment within gravel and cobble substrate, termed "embeddedness", at each potential spawning site on a four-level ordinal scale, from 1 (very low levels of fine sediment) to 4 (very high levels of fine sediment). Through dynamic segmentation and calibration (Radko 1997), we spatially linked reach-scale data to a drainage network within a geographic information system (GIS) and, using a 10-m digital elevation model (DEM), calculated stream gradients.

From the embeddedness data we calculated an embeddedness index (EI) for each surveyed reach by subtracting the proportion of spawning sites with very low embeddedness (level 1) from those with very high embeddedness (level 4). Thus, the index can range from negative 100 (all units have embeddedness value 1) to 100 (all units have embeddedness value 4). By considering only the endpoints of this scale (i.e., the extreme values of 1 and 4), we reduced the error inherent in an essentially qualitative ranking score while still allowing for maximal spread in the response variable. To determine how much information was lost by using only the endpoints of the scale, we also calculated a weighted average embeddedness for each reach that used all embeddedness values.

Analyses were restricted to reaches with at least 10 samples of embeddedness (mean = 47 samples) and a low gradient (<3%) likely to show a depositional response to sediment supply (Montgomery and Buffington 1993). For each of the 54 reaches meeting these criteria, we used the DEM and the ArcView (ESRI, Redlands, California) extension FlowZones to derive a watershed above the downstream end of each surveyed reach. We also used the DEM to derive slope classes for the watersheds.

Explanatory variables were obtained from several sources and entered into the GIS. For LULC, we used a California Vegetation data layer derived from 1994 Landsat TM with a 1-ha minimum mapping unit (California Department of Forestry Land Cover Mapping and Monitoring Program). We used aggregated LULC categories of agriculture (row crops, vineyards, and orchards), herbaceous (annual grasslands), forest (including hardwood, conifer, and mixed evergreen forests), shrubs (generally chaparral), and urban. Road density was calculated by summing the lengths of the 1:100 000 scale US Census Bureau TIGER 2000 roads and dividing that sum by the size of the area of analysis (km road·km$^{-2}$ area). For geology, we used the 1:750 000 GIS data for the Geologic Map of California (California Geographic Survey).

We quantified the amount of all LULC, geology, and road density variables within five zones of influence (Fig. 2). The "watershed" zone included all areas that drained into the downstream end of each reach. For the "local" zone, we delineated the area surrounding each reach at two different distances (30 m and 60 m) from the stream banks using the GIS buffering tool. These two distances provided essentially identical information, so subsequent analyses include only the 30-m riparian buffer. For the "upstream" zone, we buffered the riparian corridor by 30 m on each side of the stream 1 km upstream of the upper end of each surveyed reach. We then used the DEM to create a stream network upstream of each reach, with headwater channels initiating at a drainage area of 2 ha. This "network" zone was then buffered by 30 m. Our objective with this routine was to capture all drainage pathways, including intermittent and ephemeral channels, not captured by standard maps of "blue-line" streams (Hansen 2001). The contributing area for defining channel initiation is likely conservative based on field studies in the region (Montgomery and Dietrich 1988). Finally, we defined the "hillslope" zone of influence as the area that remained after subtracting the area of the network zone from the area of the watershed zone. Thus this zone evaluates the influence of land cover not directly adjacent to the densified channel network on patterns of stream sediment.

## Data analysis

We initially plotted the distribution of values for the embeddedness index across three groups of watersheds that varied in the amount of land classified in a category indicating development (i.e., either urban or agriculture): no land classified as developed, low development (1%–5% of land classified as developed), or moderate to high development (>5% developed). We then used simple linear regression to explore the relationships between single explanatory variables and the embeddedness index for the five zones of influence within all 54 watersheds. Explanatory variables with non-normal distributions were transformed to improve the homoscedasticity of residuals. For example, the proportion of zones of influence in agriculture and urban land uses were cube-root transformed. We then used stepwise regression to develop multiple regression models. We based our final model selection on the adjusted $r^2$ ($r^2_{\mathrm{adj}}$) and significance of the coefficients. We interpreted individual regression co-

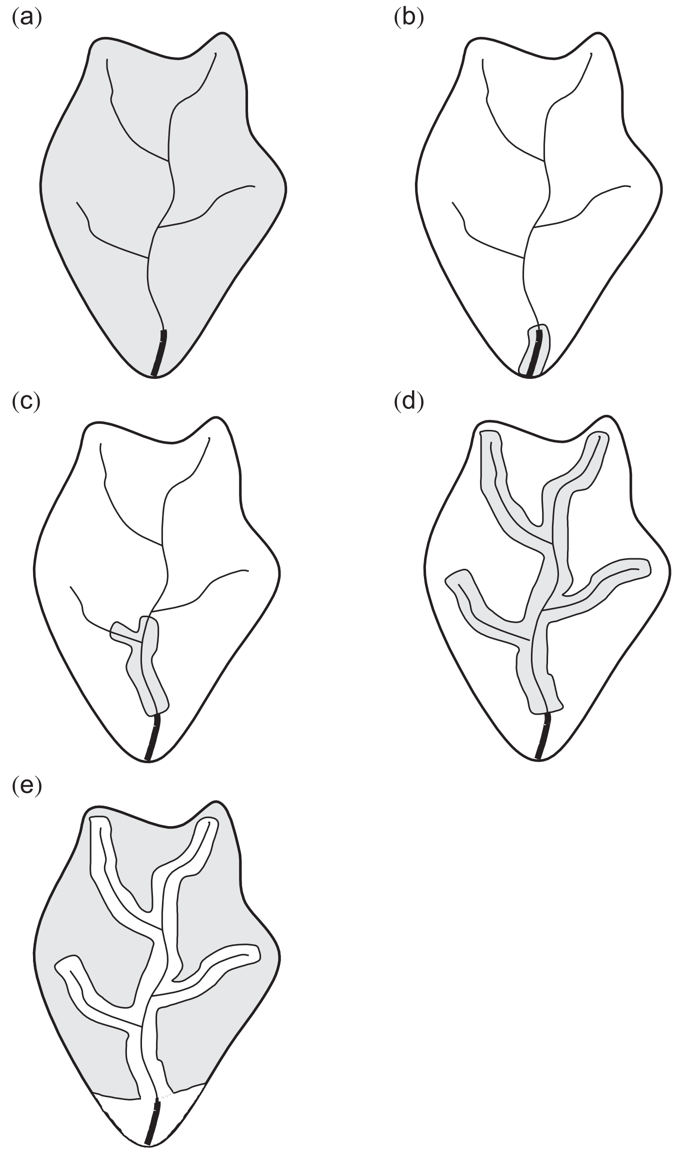



**Fig. 2.** Conceptual maps showing the five zones of influence within a hypothetical watershed: (*a*) watershed, (*b*) local, (*c*) upstream, (*d*) network, and (*e*) hillslope. The thick line represents the focal reach (i.e., the reach in which embeddedness data were collected), and the shaded area represents the portion of the watershed included in the analysis for a given zone of influence.

efficients from the multiple regression analyses with caution because moderate correlation occurred between potential predictors at the various scales.

In addition to these analyses, we evaluated the influence of size of the watershed on the predictive power of the models. We categorized the watersheds by size into three classes ($n$ = 18 per class): small watersheds (<1000 ha), intermediate watersheds (1000–3400 ha), and large watersheds (3400 – 22 000 ha). We repeated a subset of these analyses after removing those watersheds in which timber harvest is likely a major influence, because our data could not adequately address the potential influence of current and historic timber harvest (e.g., our classification for forest did not differentiate among an old-growth forest, one that had been cut 100 years ago, and one that had been cut 20 years ago). The eliminated watersheds had more than 10% of the land under a Timber Harvest Plan (THP) with the California Department of Forestry.

## Results

Explanatory variables showed moderate levels of correlation (Table 1). For example, forest cover was negatively correlated with agriculture, while road density was positively correlated with urban land cover and somewhat correlated with agricultural land cover. The proportion of sedimentary geology was positively correlated with agriculture (correlation coefficient = 0.89) and negatively correlated with forest (–0.50). Proportions of a specific land cover (e.g., agriculture) at the watershed zone of influence were highly correlated with the network (correlation coefficient >0.90) and hillslope (>0.98) zones and less strongly correlated with the local zone. Correlations between watershed and local LULC were higher within small watersheds (agriculture = 0.50, forest = 0.49) than they were for the large watersheds (agriculture = 0.35, forest = 0.17).

The three size classes of watersheds contained different distributions of LULC proportions for the land covers most associated with sediment production in the study region — urban and agricultural. The intermediate watersheds contained very little urban area and the smallest watersheds had no urban areas. The largest and smallest watersheds had somewhat similar distributions of agriculture (in terms of interquartile range and maximum), while the intermediate watersheds had very little agriculture (Fig. 3*a*). However, the distribution for road density was very similar among the three size groups (Fig. 3*b*).

The distribution of EI values varied considerably among groups of watersheds that had no area classified as developed (EI = –12 ± 7 (mean ± standard error), median = –22), those that had between 1% and 5% developed (6 ± 10, median = 0) and those that had greater than 5% developed (43 ± 9, median = 35) (Fig. 4). The stream with the highest EI in the no-development group has had recent timber harvest activity (Fig. 4*a*).

Linear regression analyses showed that the amount of agriculture and road density were consistently positively associated with EI and the amount of forest cover was consistently negatively associated with EI, across zones of influence and for different size groups of watersheds (Table 2 and Fig. 5*a*). However, LULC variables were rarely significant

**Table 1.** Coefficients of correlation between explanatory variables for the watershed zone of influence for all 54 watersheds.

| | Agriculture | Urban | Herbaceous | Shrub | Forest | Road density | Volcanic | Franciscan | Sedimentary | Low relief | High relief |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Agriculture | | 0.23 | –0.12 | –0.06 | **–0.63** | **0.52** | 0.10 | **–0.32** | **0.89** | **0.85** | **–0.78** |
| Urban | 0.23 | | **0.30** | –0.16 | **–0.40** | **0.60** | 0.10 | **–0.30** | **0.28** | **0.58** | **–0.48** |
| Herbaceous | –0.12 | **0.30** | | **–0.28** | –0.25 | 0.02 | 0.18 | 0.08 | –0.03 | 0.19 | **–0.31** |
| Shrub | –0.06 | –0.16 | **–0.28** | | **–0.52** | –0.17 | 0.20 | 0.27 | –0.19 | –0.12 | 0.06 |
| Forest | **–0.63** | **–0.40** | –0.25 | **–0.52** | | **–0.37** | **–0.33** | 0.05 | **–0.50** | **–0.70** | **0.73** |
| Road density | **0.52** | **0.60** | 0.02 | –0.17 | **–0.37** | | –0.02 | –0.15 | **0.60** | **0.69** | **–0.67** |
| Volcanic | 0.10 | 0.10 | 0.18 | 0.20 | **–0.33** | –0.02 | | **–0.39** | –0.09 | 0.22 | **–0.33** |
| Franciscan | **–0.32** | **–0.30** | 0.08 | 0.27 | 0.05 | –0.15 | **–0.39** | | –0.27 | **–0.33** | **0.28** |
| Sedimentary | **0.89** | **0.28** | –0.03 | –0.19 | **–0.50** | **0.60** | –0.09 | –0.27 | | **0.80** | **–0.72** |
| Low relief | **0.85** | **0.58** | 0.19 | –0.12 | **–0.70** | **0.69** | 0.22 | **–0.33** | **0.80** | | **–0.95** |
| High relief | **–0.78** | **–0.48** | **–0.31** | 0.06 | **0.73** | **–0.67** | **–0.33** | **0.28** | **–0.72** | **–0.95** | |

**Note:** Low relief, the proportion of the watershed with slopes <10%; high relief, the proportion of the watershed with slopes >30%. Correlation coefficients in bold are statistically significant at $p \leq 0.05$.

**Fig. 3.** Distribution of explanatory variables for three size groups of watersheds ($n$ = 18 for each; ×, median; brackets, interquartile range; △, maximum): (*a*) proportion agriculture (transformed as cube root of proportion agriculture to improve normality); (*b*) road density.

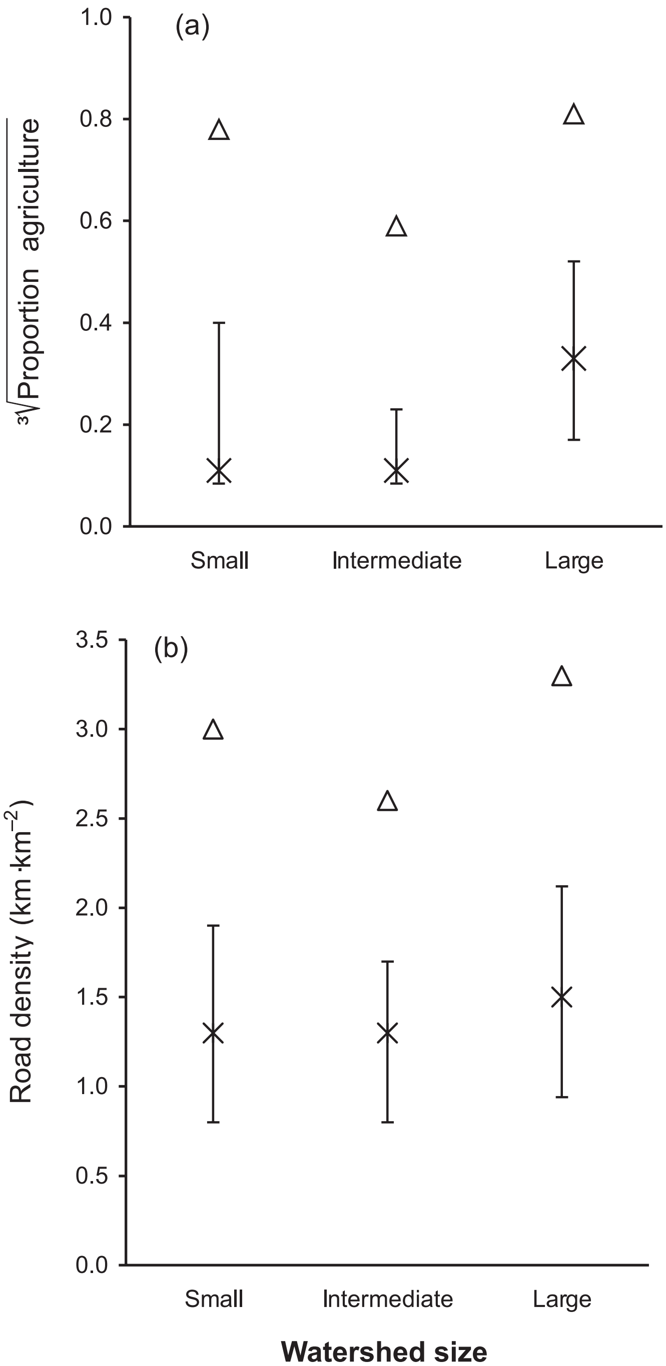
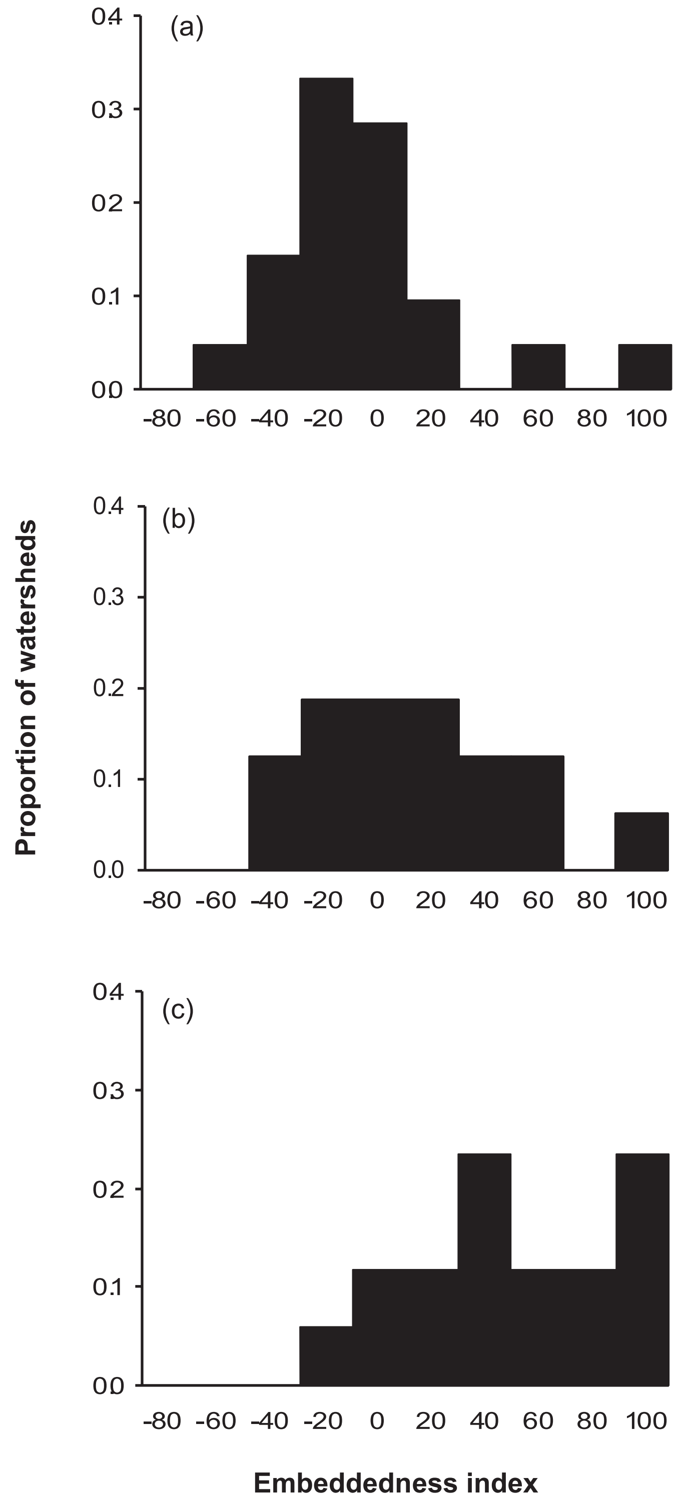



**Fig. 4.** Distributions of embeddedness index values among watersheds with (*a*) no development (defined as agricultural and urban land cover; $n$ = 21); (*b*) 1%–5% developed ($n$ = 16); and (*c*) >5% developed ($n$ = 17).




within the local zone of influence (Table 2 and Fig. 5*b*). When urban land cover had a significant relationship with EI, the relationship was always positive (Table 2). Several of these relationships were stronger among the largest watersheds than among the intermediate and smallest watersheds. For example, while agriculture was significantly positively related to EI among all three size groups, the $r^2$ for the largest watersheds was 0.53 compared with 0.21 among the intermediate watersheds and 0.35 among the smallest watersheds. When one probable outlier was removed, the explanatory power ($r^2$) for agriculture increased to 0.75 for the largest watersheds (Fig. 5*a*). While this watershed had very little intensive agriculture, it had the highest proportion

**Table 2.** Linear regression results for analyses between land use – land cover variables and embeddedness index in streams in the Russian River Basin.

| | Watershed coefficient | $r^2$ | Local coefficient | $r^2$ | Upstream coefficient | $r^2$ | Network coefficient | $r^2$ | Hillslope coefficient | $r^2$ |
|---|---|---|---|---|---|---|---|---|---|---|
| **All** | | | | | | | | | | |
| Agriculture* | 95 | 0.29 | 44 | 0.12 | 60 | 0.15 | 94 | 0.25 | 82 | 0.22 |
| Urban* | 141 | 0.25 | | | 44 | 0.07 | | | 162 | 0.23 |
| Road density | 31 | 0.31 | | | 4 | 0.09 | 26 | 0.27 | 30 | 0.29 |
| Forest | –102 | 0.21 | –61 | 0.17 | –56 | 0.15 | –99 | 0.18 | –78 | 0.13 |
| **Largest** | | | | | | | | | | |
| Agriculture* | 128 | 0.53 | | | | | 96 | 0.29 | 97 | 0.38 |
| Urban* | 162 | 0.57 | | | | | | | 178 | 0.57 |
| Road density | 108 | 0.58 | | | | | 34 | 0.51 | 41 | 0.63 |
| Forest | –175 | 0.54 | | | –68 | 0.30 | –171 | 0.54 | –150 | 0.45 |
| **Intermediate** | | | | | | | | | | |
| Agriculture* | 118 | 0.21 | | | | | 116 | 0.23 | | |
| Road density | 96 | 0.33 | | | | | | | 35 | 0.26 |
| **Smallest** | | | | | | | | | | |
| Agriculture* | 71 | 0.35 | | | 84 | 0.49 | 88 | 0.36 | 59 | 0.23 |
| Forest | –81 | 0.27 | | | | | | | | |

**Note:** Analyses were run for five within-watershed scales (i.e., zones of influence) and among four groups of watersheds: all 54 and three size groups (smallest, intermediate, and largest watersheds; $n$ = 18 per group). Only results that were significant at the $p \leq 0.05$ level are displayed. Variables denoted with an asterisk (*) were transformed to improve normality by taking the cube root of the value.

urban and highest road density of all 54 watersheds. For the most part, geological variables were not correlated with EI. However, the proportion of sedimentary geology was positively and significantly correlated with EI for the watershed, hillslope, and network zones of influence.

Comparing the multiple regression models for each zone of influence for the full set of 54 reaches, the model for watershed had the greatest explanatory power ($r^2_{\mathrm{adj}}$ = 0.42), followed by network ($r^2_{\mathrm{adj}}$ = 0.32) and hillslope ($r^2_{\mathrm{adj}}$ = 0.29; Fig. 6). The local zone of influence had the lowest explanatory power ($r^2_{\mathrm{adj}}$ = 0.07). Geological variables were not selected within the multiple regression models. Stepwise regression consistently selected proportion agriculture as a significant model term, with a positive relationship with EI, for all zones of influence with the exception of local.

The size of the watershed influenced the predictive power of the empirical models, with explanatory power decreasing from larger to smaller watersheds (Fig. 6). The largest watersheds displayed trends similar to those for all watersheds combined, with the watershed zone having the greatest explanatory power ($r^2_{\mathrm{adj}}$ = 0.73), followed closely by network ($r^2_{\mathrm{adj}}$ = 0.70) and hillslope ($r^2_{\mathrm{adj}}$ = 0.65). The upstream zone had much lower explanatory power, and the local zone had almost no explanatory power. Within these models, terms for agriculture, road density, urban, and herbaceous had positive relationships with EI, while forest and shrub had negative relationships with EI.

The smallest watersheds displayed a different trend, with the upstream zone having the greatest explanatory power ($r^2_{\mathrm{adj}}$ = 0.46), followed by network ($r^2_{\mathrm{adj}}$ = 0.38); local, watershed, and hillslope zones had relatively low explanatory power ($r^2_{\mathrm{adj}}$ ranging from 0.22 to 0.34; Fig. 6). Proportion agriculture was also selected as a significant model term, with a positive relationship with EI, for all zones of influence. Intermediate watersheds had very little spread in LULC distributions for urban and agriculture (Fig. 3*a*). Explanatory power (i.e., adjusted $r^2$) for all zones of influence was generally lower than either the smallest or largest watersheds (Fig. 6).

Repeating the watershed zone analyses after removing watersheds with >10% THP somewhat strengthened the explanatory power for the overall set of watersheds ($n$ = 39) and the largest watersheds ($n$ = 14) but did not change the explanatory power for the smallest watersheds ($n$ = 14). The two methods of summarizing embeddedness from the CDFG data (the EI and the weighted average) were highly correlated (correlation coefficient = 0.97). The analyses conducted with the weighted average as the response variable had essentially identical results in terms of explanatory power, model terms, and coefficients.

## Discussion

LULC variables were effective predictors of the levels of embeddedness of spawning substrate in streams within the Russian River Basin. Considering all 54 watersheds, LULC had the greatest explanatory power within the watershed, network, and hillslope zones of influence, with less explanatory power for the local and upstream zones. However, watershed size appears to have some influence on the relative strengths of these relationships. The strongest overall relationships between LULC and embeddedness were found among the largest watersheds at the watershed, network, and hillslope zones (with $r^2$ between 0.65 and 0.73). Conversely, among the largest watersheds, the local zone had essentially no explanatory power. The next strongest relationships between LULC and EI were found among the smallest watersheds at the network and upstream scales. The intermediate watersheds had very low levels of LULC classes generally associated with sediment production in the study region (agriculture and urban), and explanatory power for analyses

**Fig. 5.** (*a*) The embeddedness index plotted against the proportion of watershed in agriculture for the 18 largest watersheds; the arrow indicates a watershed that is less than 33% forested with 18% urban and the highest road density of the 54 watersheds. (*b*) The embeddedness index plotted against the proportion of the local riparian corridor in agriculture for the 18 largest watersheds. The proportion of agriculture in both zones of influence was transformed as a cube root to improve normality.

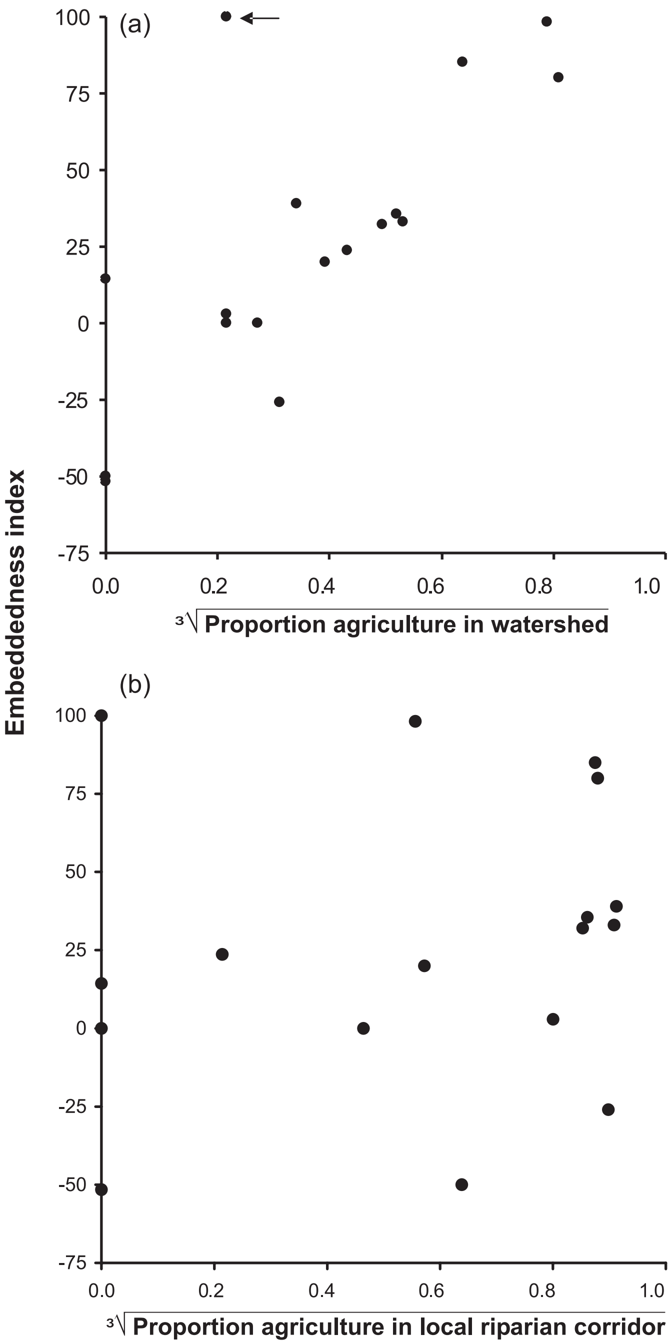
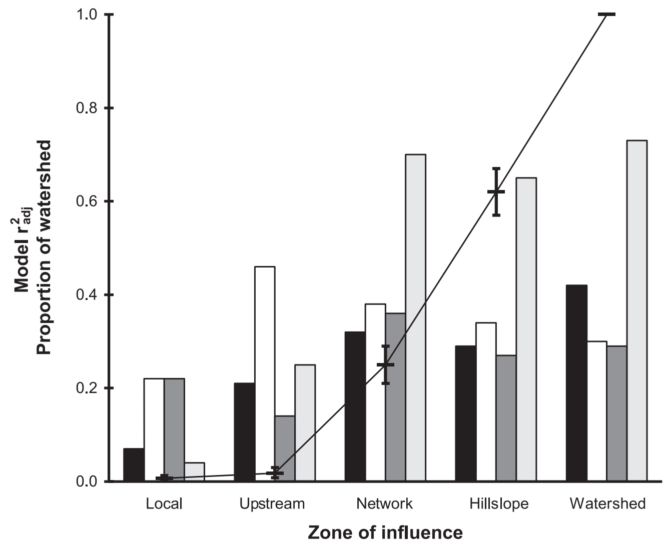



**Fig. 6.** Explanatory power ($r^2_{adj}$) of models focused on five zones of influence for four samples of watersheds: all 54 watersheds (solid bars), smallest (open bars), intermediate (darker shaded bars), and largest (lighter shaded bars) watersheds. The line reflects the proportion of the watershed's total area that is encompassed within a given zone of influence (median ± 25th and 75th percentiles (brackets)).




among these watersheds was generally lower than that for either the largest or smallest watersheds.

Results from our analyses showed that across multiple models and scales, LULC categories for agriculture, urban, and road density were highly significant model terms, explained the most variation in EI, and were consistently positively correlated with embeddedness, while forest cover was always negatively correlated. Agriculture can lead to significantly higher rates of sediment production, even on moderate slopes, because of the increased amount of bare soil exposed to the erosive power of raindrops and sheet wash (Dunne and Leopold 1978; Chang et al. 1982; Pimentel and Kounang 1998). Agriculture can also lead to higher rates of runoff (Chang et al. 1982), which can then increase sediment load through incision and bank erosion (Kuhnle et al. 1998). Similarly, urban land cover can increase peak runoff and, consequently, channel erosion (Trimble 1997; Pizzuto et al. 2000), in addition to the large amounts of fine sediment produced during periods of construction (Dunne and Leopold 1978). Numerous studies have linked roads with the production of sediment, particularly in areas with rugged topography (Swanson and Dyrness 1975; Montgomery 1994; Jones et al. 2000).

Our results suggest that increased sediment produced directly or indirectly from agricultural and urban areas and roads may be one mechanism by which these land uses degrade stream habitat and potentially influence salmonid abundance and recovery. In a study relating fish abundance with watershed characteristics, Bradford and Irvine (2000) reported that agricultural land use was associated with declines in coho salmon populations within 40 tributary watersheds of the Thompson River, British Columbia. Similarly, Pess et al. (2002) showed that coho salmon abundance in the Snohomish Basin in Washington was negatively correlated with the percentage of watershed in agriculture, urban development, and roads. The abundance of juvenile chinook salmon was also negatively correlated with road density in a study in Idaho (Thompson and Lee 2000). Sutherland et al. (2002) reported that increased fine sediment from disturbed watersheds in the southern Appalachians altered stream fish assemblages by reducing the abundance of species that require clean gravel and cobble for spawning. Together, this research suggests that increased sediment from agriculture, urban development, and roads can alter fish abundance and assemblages.

Timber harvest is another land use frequently associated with fine-sediment delivery to streams (Everest et al. 1987; Platts et al. 1989; Lewis 1998). Although timber harvesting is less common in this region than in conifer-dominated basins of the Pacific Northwest, several watersheds in this study currently or historically contained timber harvests. However, the LULC data we used in this study could not adequately characterize the role of forestry activities (e.g., timing, type, or extent of harvests). Timber harvest may have contributed to some of the unexplained variation in the models, as the measure of $r^2$ generally improved after removing 14 watersheds likely affected by historical or current forestry activities.

In our analysis, different size groups of watersheds displayed different patterns of explanatory power between LULC and EI across different zones of influence. These different trends likely reflect differences in (*i*) spatial distribution of land-use type, (*ii*) spatially dependent processes, and (*iii*) data resolution. For example, the relatively low explanatory power of models for the intermediate-sized watersheds is likely due to the small amounts of urban and agricultural developments within them. Because there was so little variation in these explanatory variables, it is not surprising that LULC variables could explain relatively little variation in EI. Therefore, it is difficult to draw generalizations about watershed scale from the intermediate watersheds.

The strongest relationships overall occurred among the largest watersheds, which may be in part because these watersheds had the broadest distribution of values for explanatory variables such as urban, agriculture, and forest. However, the distributions for road density were very similar among the three size groups of watersheds, and the large watersheds had a much stronger relationship between road density and EI ($r^2 = 0.58$) than did the intermediate ($r^2 = 0.33$) and small ($r^2 = 0.04$) watersheds, suggesting that factors in addition to the distribution of explanatory variables are responsible for the differences in explanatory power based on watershed size.

Much of the difference in explanatory power between the watershed size groups is likely due to the natural tendency of smaller watersheds to have more variable sediment fluxes than large watersheds (Benda and Dunne 1997). In mountainous watersheds, landslides and other mass-wasting events, including natural events and those induced by roads, timber harvest, or other activities, are often responsible for a large proportion of the sediment supply to a channel network (Dietrich and Dunne 1978; Montgomery et al. 2000). Mass wasting is stochastic in both space and time (Kelsey 1980) and, within small watersheds, can be a rare event. Within watersheds of progressively larger drainage area (i.e., aggregating numerous small watersheds), the frequency and number of landslides within the basin increases correspondingly. Larger watersheds integrate the stochastic pulses of sediment occurring within their smaller subcatchments and thus dampen the variability of sediment fluxes through reaches that drain large areas. Because of this process of integrating sediment inputs, larger watersheds are more likely to show a land-use signal resulting from the different rates of runoff and sediment produced by different land covers (Wark and Keller 1963; Dunne and Leopold 1978; Chang et al. 1982).

Further, it is likely that small watersheds were more strongly influenced by conditions and processes that we could not detect with the resolution of our data. For example, a small watershed (e.g., <1000 ha) that is mostly forested may contain an unpaved road network throughout the watershed that is particularly problematic in terms of sediment production. It is less likely that a large watershed (e.g., 10 000 ha) that is primarily forested would encompass a similarly extensive (relative to watershed area) problematic road network. Thus, because forested areas generally produce less sediment than other land classes (Chang et al. 1982; Sutherland et al. 2002), the larger watershed will show a sediment signal more influenced by its overall land cover. Additionally, it is much more likely that a discreet land-use conversion event (such as timber harvest or clearing an oak woodland for a vineyard) will generate a more readily detectable sediment signal in a small watershed than in a large one. Because our LULC data are snapshots in time, they may be more appropriate for capturing the coarse-scale and integrative relationship between LULC and sediment in larger watersheds.

Results from our study also showed that LULC variables within several different zones of influence were good predictors of embeddedness, with variations according to watershed size. Strayer et al. (2003) hypothesized that the spatial arrangement of patches was more critical within small watersheds (<1000 ha) than it was for large watersheds. Conversely, variables that integrate information, such as proportion of a watershed in agriculture, will have greater explanatory power for large watersheds, which integrate inputs from numerous small watersheds. Consistent with this hypothesis, we found the strongest relationships between LULC and EI among the smallest watersheds were within the network and upstream zones. These zones, which are directly adjacent to the drainage network or main stem, provide more information about the spatial arrangement of patches than do the watershed or hillslope zones. Among the large watersheds, the network zone had very high explanatory power, but so did the hillslope and watershed zones.

This potential effect of scale (watershed size) on explanatory power may partly explain past conflicting results and should also be considered in any similar analysis. For example, Jones et al. (1999) and Sponseller et al. (2001) concluded that riparian land use (a scale which provides information on spatial arrangement) had stronger influence on in-stream variables than did catchment-scale land use (an integrative scale); the watersheds in these studies averaged 535 and 2366 ha, respectively. Conversely, studies by Roth et al. (1996) and Hunsaker and Levine (1995) showed that the watershed characteristics, rather than the riparian corridor characteristics, had greater influence on in-stream habitat; these watersheds averaged 6804 and 93 000 ha, respectively. Our comparative approach demonstrates that the size of the study watersheds may be one reason for differences observed between these studies.

In this study, we found that the local zone generally had minimal predictive power for embeddedness. The moderate explanatory power for the local zone among small watersheds may be due, in part, to the higher correlation between

LULC classes (e.g., forest and agriculture) at the local and watershed zones for small watersheds than for large watersheds. The general lack of influence of the local zone is not surprising, given that it represents a very small proportion of the total watershed and the surveyed reaches are receiving runoff from hundreds to thousands of hectares. Because riparian land cover had little relationship to embeddedness, it is unlikely that reach-scale riparian restoration will improve spawning conditions within the lower reaches of watersheds if conditions within the rest of the watershed remain unchanged. However, we do want to note that it is difficult to determine which zone of influence has the strongest predictive power for embeddedness, in large part because LULC in the zones of watershed, hillslope, and network are so highly correlated.

Because the LULC variables were derived from one moment in time, this analysis cannot account for the timing of land-use conversion or capture the legacies of historical land use, which can continue to exert influences on streams for decades (Harding et al. 1998). Historical legacies and the timing of conversions are undoubtedly responsible for a large portion of the unexplained variation in the models. Studies, such as this one, that examine relationships between current land-use patterns and in-stream variables must be cautious to not infer causation from correlation. For example, we do not know the distribution of embeddedness values within Russian River watersheds prior to any anthropogenic changes in land cover. However, comparing the distributions of EI within watersheds with no development with those more than 5% developed provides strong support for the role of anthropogenic land-cover conversion in increasing sediment levels in Russian River tributary streams. Although this study's approach cannot explain the mechanisms by which this happened, the mechanisms by which changes in land cover increase sediment production are well established at smaller scales (e.g., plots, small experimental watersheds) (Dunne and Leopold 1978; Chang and Tsai 1991; Battany and Grismer 2000). Combining empirical approaches with future mechanistic research (both experimental and modeling) will further improve our understanding of the linkages between land use and in-stream habitat (Strayer et al. 2003).

Our data are the first to relate patterns of fine sediments in streams to patterns of land use in Mediterranean-climate watersheds in California, and these data suggest that agricultural land use is correlated with elevated levels of fine sediment. Over the past decade, vineyards in the Russian River and Napa River basins have expanded onto hillslopes because of limited land availability in valley bottoms (Merenlender 2000). A recent study projects that 80% of future vineyards in Napa County will be planted on hillslopes (Napa County RCD 1997). Hillside vineyards in this region can produce annual soil loss ranging from 5 to 50 $t \cdot ha^{-1}$ (Battany and Grismer 2000). Battany and Grismer (2000), working in Napa County, found that soil cover was the primary factor affecting erosion rates from hillslope vineyards, with slope as a secondary factor. This observation emphasizes the potentially important role that management practices can play in reducing the impacts from agricultural land use.

In conclusion, the results of our study suggest that watershed-scale patterns of land use, including both the land adjacent to the entire upstream channel network and the surrounding hillslopes, are generally the best predictors of stream sediment conditions. Local land cover (i.e., the adjacent riparian corridor) had little relationship to embeddedness. Much attention and resources have been spent on piecemeal stream restoration and sediment control efforts at the local scale (e.g., bank stabilization). Our data indicate that the effects of such localized efforts will be overwhelmed by processes operating at larger scales and, thus, have little influence on spawning conditions. Rather, to improve spawning gravels, restoration efforts should emphasize protecting riparian corridors throughout entire watersheds and promote programs or policies that ameliorate the influences of roads and agricultural land use. However, even watersheds with relatively low levels of development (e.g., 5% of a watershed) had relatively high embeddedness scores, suggesting a limit to the improvements that restoration programs can hope to achieve. The landscape-scale approach of this study emphasizes the overarching importance of large-scale land-use patterns. With such an approach, managers and policy-makers can identify priority areas for protection and restoration and, when combined with projections of future land-use change, identify particularly vulnerable watersheds.

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